PB84-201649

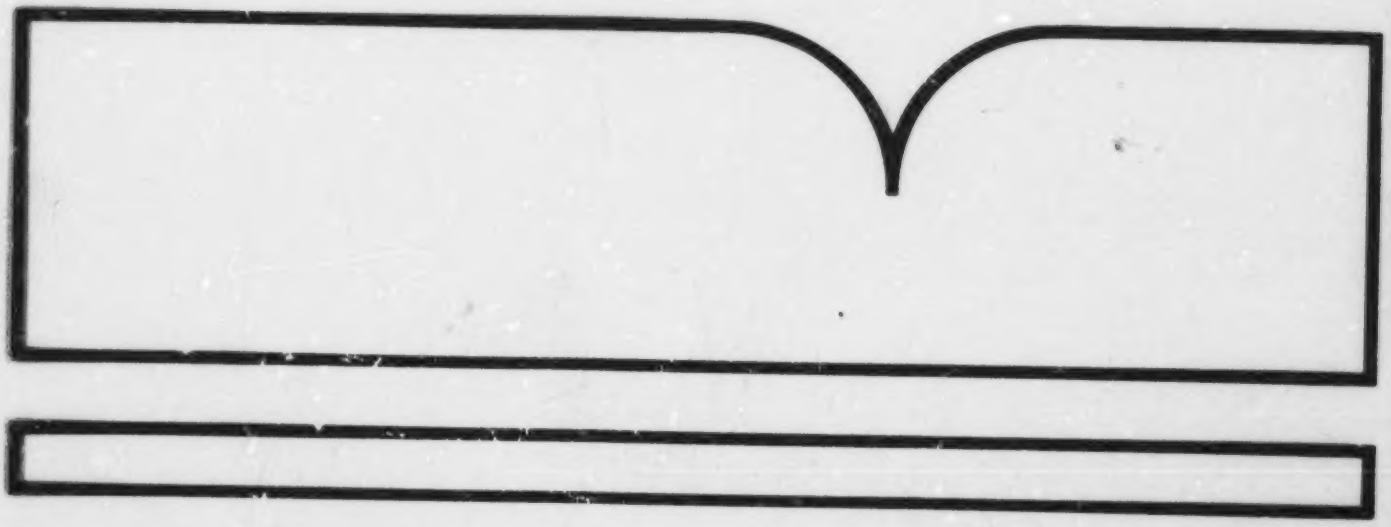
U.S./Canada Aquatic Impacts Assessment Integration of Experimental Studies Monitoring and Modeling of Acid Deposition Effects

Institute of Ecology, Indianapolis, IN

Prepared for

Environmental Research Lab.
Duluth, MN

Jun 84



U.S. Department of Commerce
National Technical Information Service
NTIS

PB84-201649

EPA-600/D-84-148
June 1984

THE U.S./CANADA AQUATIC IMPACTS ASSESSMENT: INTEGRATION OF EXPERIMENTAL STUDIES, MONITORING AND MODELING OF ACID DEPOSITION EFFECTS

by

Orie L. Loucks
The Institute of Ecology
4600 Sunset Avenue
Indianapolis, Indiana 46208

Gary E. Glass
Environmental Research Laboratory
U.S. Environmental Protection Agency
Duluth, Minnesota 55804

ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
DULUTH, MN 55804

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

TECHNICAL REPORT DATA (Please read Instructions on the reverse before completing)		
1. REPORT NO. EPA-600/D-84-148	2.	3. RECIPIENT'S ACCESSION NO. PB84 201649
4. TITLE AND SUBTITLE The U.S./Canada Aquatic Impacts Assessment: Integration of Experimental Studies, Monitoring and Modeling of Acidic Deposition Effects		5. REPORT DATE June 1984 6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) O. L. Loucks* and G. E. Glass**		8. PERFORMING ORGANIZATION REPORT NO.
9. PERFORMING ORGANIZATION NAME AND ADDRESS * The Institute of Ecology, 4600 Sunset Ave. Indianapolis, IN 46208 **Environmental Research Lab., U.S. EPA Duluth, MN 55804		10. PROGRAM ELEMENT NO. 11. CONTRACT/GRANT NO.
12. SPONSORING AGENCY NAME AND ADDRESS Environmental Research Laboratory Office of Research and Development U.S. Environmental Protection Agency Duluth, MN 55804		13. TYPE OF REPORT AND PERIOD COVERED 14. SPONSORING AGENCY CODE EPA-600/03
15. SUPPLEMENTARY NOTES		

16. ABSTRACT

Assessments of pollutant effects on living organisms usually have used dose-response relationships (models) based on data for one pollutant and one species at a time, for a relatively short period of exposure, under relatively controlled conditions. Such studies are necessary, but even literature reviews of the results provide only limited insight into the response of regional resource systems where there are multiple pollutant impacts. For the assessment planned under the 1980 U.S./Canada Memorandum of Intent on long-distance pollutants a Work Group was established specifically to evaluate effects from the apparent elevated levels of pollutants transported long distances. This paper reports on the use of both qualitative comprehensive models and quantitative empirical models as methodologies for the Aquatic Impacts Assessment process in Phases I and II of that study (1).

17. KEY WORDS AND DOCUMENT ANALYSIS		
a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
18. DISTRIBUTION STATEMENT Release to Public	19. SECURITY CLASS (This Report) Unclassified	21. NO. OF PAGES 24
	20. SECURITY CLASS (This page) Unclassified	22. PRICE

EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE

NOTICE

This document has been reviewed in accordance with U.S. Environmental Protection Agency policy and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

INTRODUCTION

Past assessments of pollutant effects on living organisms have generally used dose-response relationships (models) based on data for one pollutant and one species at a time over a relatively short period of exposure under relatively controlled conditions. Such studies are necessary, but even literature reviews of the results provide only limited insight into the response of regional resource systems where there are multiple pollutant impacts. For the assessment planned under the 1980 U.S./Canada Memorandum of Intent on long-distance pollutants, a Work Group was established specifically to evaluate effects from the apparent elevated levels of the several pollutants known to be transported long distances. This paper reports on the use of both qualitative comprehensive models and quantitative empirical models as methodologies for the Aquatic Impacts Assessment process in Phases I and II of that study[1] (1).

APPROACH

One must consider, first, the full spectrum of research questions that face scientists analyzing the effects of acidic deposition on whole drainage basins or landscapes. The various manipulations of agricultural lands and forests have altered both the water budget and the nutrient budget of surface layers, which, in turn, may have affected the water table and base-flow to springs, streams, and adjacent lakes. Thus, a major step in focusing the research questions was to develop a mental working model of the transport and

---

[1] The contributions of Dr. Thomas Brydges of the Ontario Ministry of the Environment, and of other U.S. and Canadian participants in the study are hereby gratefully acknowledged.

Reproduced from best available copy.

interactions involved in materials being exchanged between air, land and water. Such models have been available for several years (2). The models also have documented the processes by which watersheds respond to acidic deposition, relating transfers between air and land, and land and water to subsequent effects. The biotic and abiotic transfers and interactions that link the three media together can be shown as descriptive flow or compartment transfer relationships (Figure 1). Implicit here is a relatively formal systems approach as a guide to the critical evaluation of the otherwise fragmented information base available at the time. This qualitative model of the relationships between the subsystems and the potential environmental control of the whole system is, in effect, a suite of hypotheses that must, ultimately, be tested sequentially.

The descriptive model in Figure 1 can be thought of as a general statement of the relationships implicit in how lakes (and inlet and outlet streams) respond to atmospheric acid inputs. As such, it represents theory, but it also represents the general outline of a report evaluating the significance of acidic deposition for aquatic resources. An analysis of the principal activity areas in Figure 1 leads to identification of four sections covering the major components of an applications-oriented systems model:

I. Distribution and variability of the input driving variables (pollutant deposition);

II. The processes in the responding system (watershed biogeochemistry), and observed response relations (time-trend monitoring);

III. Quantification of biological response relationships (biotic impacts);

IV. Quantification of predictive/summarative models.

Reproduced from best available copy.

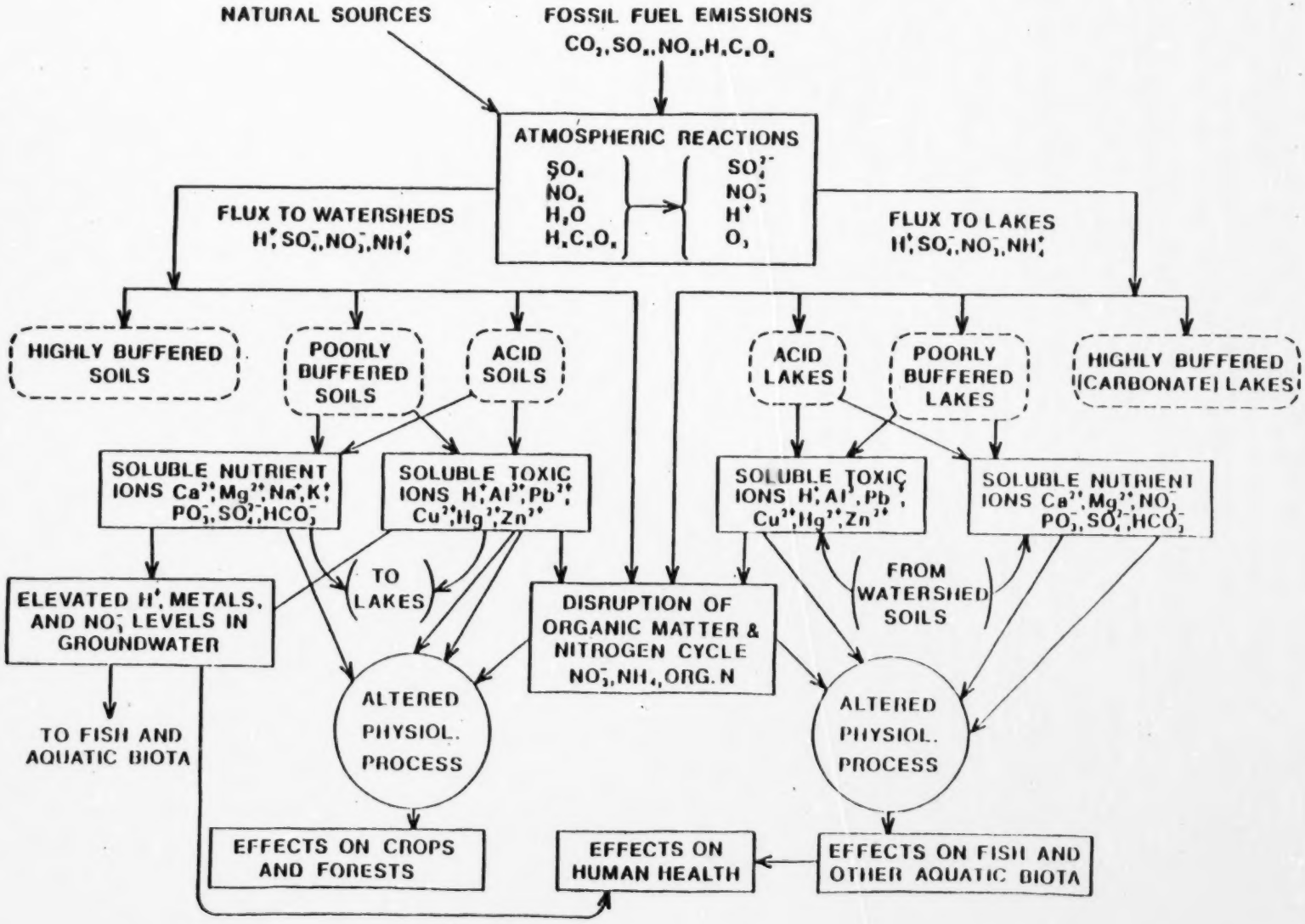


Figure 1. Flow diagram showing system linkages for acid precipitation formation, deposition and effects as a consequence of nitrogen and sulfur oxide emissions from fossil fuel combustion. From Loucks (3).

The expanded general outline adopted for the aquatic effects section of the Phase II report (1), as directed by the descriptive model, is shown in Table 1. The theory and the model, despite their limitations, provide paradigms and testable hypothesis-sets that can be addressed in the various sections of an assessment. Thus, the objective of the aquatic impacts sub-group in this assessment was to seek a broad, balanced review of the effects of acidification on the production processes of aquatic ecosystems, while at the same time seeking to summarize knowledge about the whole system.

Approaches utilizing integration and synthesis have been widely used in resource alteration problems where strong traditions in quantitative science exist (4, 5). The relationships implicit in this model can incorporate a multi-regional research summary (or report outline), so as to focus research results on critical questions in a linked-system context (6). Thus, the specific goal for the U.S./Canada assessment was to consider all evidence available on the relationships implicit in the model.

CONTRIBUTIONS OF THE ASSESSMENT PROCESS

Most recent assessments of potential effects from acidic precipitation in the eastern U.S. and Canada (7, 8) conclude that much more research is needed before we can know with any confidence that natural resources may be truly at risk. The assessments to date have generally concluded that the low pH of rainfall in the eastern part of North America is unusual for that area, and that its distribution is associated with the principal man-made sources of nitrogen and sulfur oxide emissions. Thus, while many of these scientific uncertainties still exist, they are imbedded in the complex functioning of closely-linked land, air, and water systems, and their associated biota.

Reproduced from best available copy.

Table 1. Outline of an Assessment of Responses in Watershed Systems to Acidic Deposition

Class or Variable	Component Variable or Process
Ia The Hydrologic Cycle	
Ib Atmospheric Input, Transport, and Deposition of Pollutants	
	Emissions of Pollutants to the Atmosphere
	Atmospheric Transport of Pollutants
	Atmospheric Removal Processes
	Alteration of Precipitation Quality
IIa Element Fluxes and Geochemical Alterations of Watersheds	
	The Hydrogen Ion
	Nitrate and Ammonium Ions
	Sulfate
	Aluminum and Other Metals
	Effects of Natural Organic Acids
	Acidity Contribution
	Buffering Capacity
	Cation and Anion Budgets
	Element Budgets in Hubbard Brook
	Element Budgets in Canada
	Regional Responses in Lake Chemistry
IIb Alteration of Surface Water Quality	Effects of Forest Manipulation
	Present Status of Sensitive Aquatic Systems
	Time-Trends in Surface Water Alterations
IIIa Alteration of Biotic Components	Seasonal and Episodic pH Depression
	Effects on Algae
	Effects on Aquatic Macrophytes
	Effects on Zooplankton
	Effects on Aquatic Macroinvertebrates
	Effects on Microorganisms
	Effects on Amphibians
	Effects of Low pH on Fish
	Studies of Aluminum Effects on Fish
	Impacts on Fisheries in Canada and the U.S.
	Loss of Genetically Unique Fish Stocks
	Mercury in Fish
	Effects on Birds and Mammals
IIIb Evidence for Irreversible Impacts	Effects on Unique Ecosystems
	Depletion of Buffering Capacity
	Soil Cation and Nutrient Depletion
IV Sulfate Loading Tolerances	Loss of Gene Pools (Unique Components)
	The Relative Significance of S and N
	Methodologies for Estimating Target Loadings
	1. Almer/Dickson Relationship
	2. Henriksen Nomograph
	3. Cation Denudation Rate (CDR) Model
	4. Short-Term or Episodic Effects
	Summary of Loadings and Target Loadings

Using Figure 1 for the systematic examination of evidence on each component of the material flows through watersheds, the U.S./Canada Phase II study reached the following major conclusions.

Patterns Among Input Driving Variables: Numerous studies have shown that elevated amounts of wet and dry deposition of transported acidic substances (the dominant input variables) are currently being observed over much of eastern North America (9). The patterns of deposition for SO_4^{2-} is shown in Figure 2. Within this half-continent are large areas where the surface soil material and bedrock types have little acid neutralizing capacity for the acidic inputs (10). These are identified as "potentially sensitive". The three most susceptible regions in the eastern one-half of the United States, based on the acid-neutralizing constituents of bedrock and soils, are the Northeast, the Appalachian Mountain regions and the Minnesota-Wisconsin-Michigan highlands (1, 3, 10).

The significance of each constituent of this regional deposition, i.e., dry vs. wet, and aspects of the relative importance of S and N compounds, are still being evaluated, but results reviewed for the Phase II report[1] indicate wet and dry deposition of sulfur compounds are of about equal importance on an annual basis (1, 10). Dry deposition of SO_2 and SO_4^{2-} is of much less importance than wet deposition at sites remote from the principal source areas (11).

Regional Patterns in Water Quality: The potential impacts from the deposition of acidifying substances (sulfur and nitrogen oxides, ammonium and strong acids) on water quality, and on the aquatic ecosystems, are becoming relatively quantified. Several examples have been documented where changes in water quality can now be attributed with reasonable confidence to atmospheric acidic inputs (1, 10). This supports the hypothesis that the

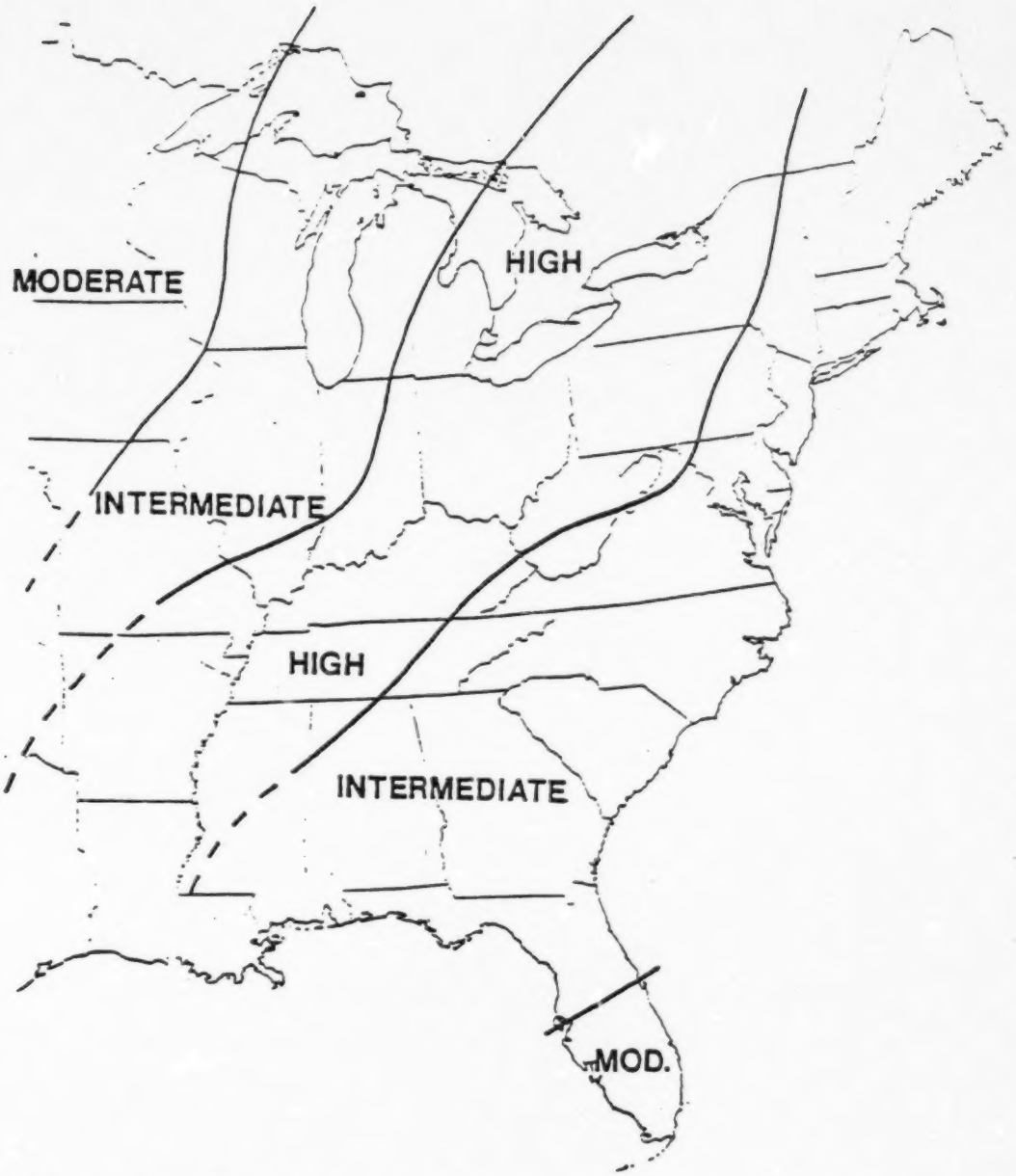


Figure 2. Distribution of SO_4^{2-} loadings in 1979-80 as an estimator of total cumulative dose: High (35+ kg/ha·yr), Int. (20-35), Mod. (10-20), and Low (<10). Based on NADP data[9,10].

atmospheric chemical inputs are the driving variables producing the water quality responses being observed. An eastern North American cross-section of data on excess sulfate in surface waters, considered in relation to the measured wet deposition of sulfate, provides strong support for the view that it is the atmospheric deposition of sulfate which accounts for elevated sulfate levels in surface waters (1, 3).

Because of the low availability of Ca and Mg and other acid neutralizing elements in the soils and waters of these regions, high SO_4^{2-} concentrations in the surface waters has led to greatly depressed concentrations of bicarbonate alkalinity (HCO_3^-). Of 1,527 lakes surveyed in Ontario, 18% were classified as extremely sensitive, i.e., having total alkalinity (largely HCO_3^-) of 0 to 40 microequivalents per litre (ueq/l). An additional 43% were considered moderately-sensitive (1). Alkalinity data obtained for the surface waters of sensitive regions in the eastern U.S., as summarized in Table 2 (1), show 38 to 81% of lakes and streams are moderately sensitive to acidified alkalinities (from less than 0 to 200 ueq/l). Thirteen to 48% of midwestern lakes and streams are in these categories.

A downward trend in pH over time is evident from several studies available to investigators (1). High elevation lakes in the Adirondacks have shown a marked pH decline over a 40-year period. Mountain streams in Virginia and North Carolina have shown changes over the past two decades. A 17-year trend toward acidification of headwater streams has been observed in New Jersey. Results from northeastern Minnesota, where acidic deposition is relatively low, show a depression in stream pH of almost 1 full pH unit during the spring snowmelt period (9).

Table 2. Percent of Lakes and Streams in Various Sensitivity Classes, Classified for Six Locations in the Eastern U.S. and Four in the Upper Midwest (1)

Region	Percentage of Lakes and Streams Sampled					Total Number Lakes/Streams Sampled
	Acidified (≤0 μeq/l)	Extreme Sensitivity (0-40 μeq/l)	Moderate Sensitivity (40-199 μeq/l)	Low Sensitivity (200-499 μeq/l)	Not Sensitive (≥500 μeq/l)	
Lakes						
Adirondack[1]	23	17	40	15	5	432
Northern Wisconsin[2]	3	20	19	32	26	276
Northern Minnesota[3]			48	35	17	85
Streams						
N.W. Pennsylvania[4]		22	33	34	11	18
E. Pennsylvania[4]		19	19	31	31	26
Connecticut[5]	4	4	48	44		23
Maine[5]		6	75	19		16
NC/VA[6]		5	59	33	3	38
Jackson Co. (Wis.)[7]		8	35	18	39	80
Oneida Co. (Wis.)[8]		2	11	33	54	195

[1]Pfeiffer and Festa (12); [2]Eilers et al. (13); [3]Glass and Loucks (14); [4]Arnold (15, as cited in 1); [5]Haines (16); [6]Hendrey et al. (17); [7]Klick and Treinen (18); [8]Andrews and Threinen (19).

The question of the significance of nitrate deposition for explaining the responses attributed to acid inputs is still being evaluated. Most of the nitric acid input appears to be assimilated (often up to 90%) by the terrestrial portion of watersheds (1, 10). At the same time, where SO_4^{2-} deposition is large, biological interactions with sulfur are small in comparison to the watershed inputs and outputs. Thus, for watersheds of low SO_4^{2-} absorption in the northeast, SO_4^{2-} is viewed as a relatively conservative anion (1) and, therefore, likely to show a direct relationship between the SO_4^{2-} concentrations observed in precipitation and those observed in lakes and streams (1, 10, 20).

On balance, the evidence available leads to the conclusion that recent increases in the acidity of surface water systems are most probably the result of increases in the dominant system forcing function, the sulfate ion and associated acidity. Although nitric acid has been observed to contribute some acidity in surface waters during snowmelt in certain locations, there is little theoretical or empirical evidence for anticipating appreciable reduction in lake or stream acidification from a reduction of nitric acid inputs. Conversely, a reduction in sulfate loading would appear likely to reduce concentrations in the system response components, the observed sulfate concentrations in surface waters.

Alteration of Aquatic Biota: Many changes in aquatic life have been linked to the alteration of surface waters cited above (1, 3, 10, 20). Causal relationships appear to have been established with depressed pH for some species, but more often the observed changes in biota are merely correlated with changes in pH. Acidification often has been accompanied by decreases in species diversity and changes in species dominance (3, 21), but acidification also results in changes in the makeup, size and metabolism of

Reproduced from best available copy.

plankton communities and, therefore, in the general structure of aquatic food chains.

Fish populations, in particular, appear to have been lost from some lakes in the Adirondacks and Ontario over the period of the past 40 years (1, 3, 20). Some lakes appear to be experiencing changes in fish populations now, while still others are in a healthy condition. Many species of amphibians (salamanders and frogs) breed in temporary pools susceptible to pH depression when accumulated acid flushes rapidly during spring or fall storms. Thus, there is a danger that they may become locally extinct. Their significance in the food chain may hold important implications for other wildlife (1, 3, 21, 21).

Evaluation of Empirical Dose/Response Models: In contrast to the assessment of step-by-step linkages within the watershed model, a number of studies have tried simply to evaluate input/output relationships for the watershed as a whole. These must be referred to as empirical models, as opposed to the more mechanistic models described above. Several different aproaches are available for examining empirical relationships between measures of acid loading and the response of aquatic ecosystems (3, 22, 23, 24). Although these models are all under active development, they have advanced to the point where comparative assessments are possible.

One model, based on Scandinavian data (Figure 3), indicates that annual sulfate loading of less than 15 to 17 kg/ha would be likely to degrade "moderately sensitive" lakes (22). The most sensitive lakes and streams appear likely to be on the borderline of potential effects at a long-term annual sulfate loading rate greater than 9 to 12 kg/ha.

A second model (23), using the Henriksen regression relationships for lake acidification, suggests that precipitation pH of 4.5, and lakewater

Reproduced from best available copy.

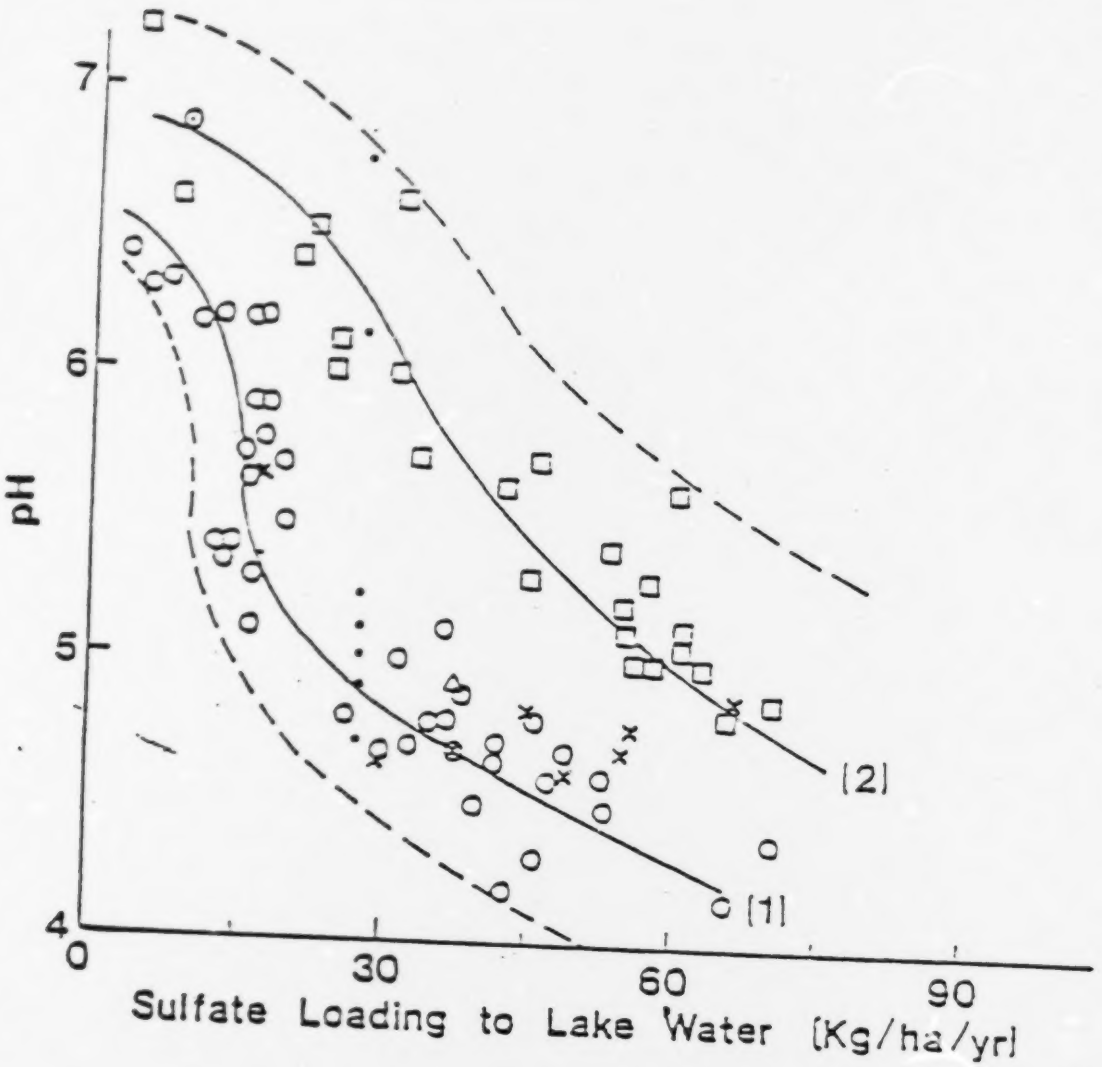


Figure 3. Effects of various sulfate loading rates on lake pH for lakes in very sensitive (circles) and somewhat less sensitive (squares) surroundings in Sweden. Added points are for: (·) Florida (Crisman and Brezonik 25); (o) Como Creek (Lewis and Grant 26); (Δ) Hubbard Brook (Likens et al. 27); (x) Norway (Wright and Snekvik 28); modified from Almer et al. (22).

SO_4^{2-} concentration of 60 ueq/l, are the maximum tolerable for lake waters with 50 ueq Ca/l or less. These lake concentrations of sulfate convert to a precipitation sulfate concentration of about 40 ueq/l. Highly sensitive lakes and streams appear to be relatively unaffected in Norway at precipitation sulfate concentrations of 21 ueq/l.

Application of a third model (24), based on estimates of cation denudation for rivers in eastern Canada, leads to an estimate of loading tolerances consistent with these findings.

A fourth model (3) uses daily monitoring of stream pH in eastern North America and data on physiological sensitivity of early life stages of fish to elevated concentrations of H^+ and Al^{3+} during flushing events (snowmelt or heavy rain). This model indicates that a sulfat e-loading of 5 to 7 kg ha^{-1} yr^{-1} produces a critical pH depression (ΔpH in the range of 0.7 to 1.0) for streams of intermediate alkalinity in sensitive areas. A loading threshold of 7 kg SO_4 ha^{-1} yr^{-1} converts to about 21 ueq SO_4/l (assuming 70 cm/yr precipitation).

A comparison of the existing depostion patterns and the indicated "no effect" or "moderate effect" tolerances is presented in Table 3 (from 1). Although the indicated dose/response relationships among these models are relatively consistent, the estimates still are preliminary. Nevertheless, the results indicate reductions in deposition in northeastern U.S. on the order of 65 to 80 percent could be required to meet the lower tolerance level.

Regardless of the model used to determine threshold watershed tolerances to acidic inputs, i.e. loading tolerances, certain variables lie outside the purview of effects modeling. One is the number of lakes placed at risk by a given "loading tolerance" that society may choose not to protect. Lake

Table 3. Observed Sulfate Loadings and Indicated Tolerances Predicted By Empirical Models (1)

Source	Loadings as Total Deposition (kg SO_4 ha^{-1} yr^{-1})	Loadings as Wet/(Bulk) Deposition	Comments
Observed Loadings			
Observed at Hubbard Brook, New Hampshire	55[1,2]	(35)[1]	Acidified lakes in the area.
Observed at Panther Lake, Adirondacks, New York	62[2]	38[2]	Acidified lakes in the area.
Observed at Dorset, Ontario	42[3]	30[3]	pH depression causing fish kills.
Observed at Kejimkujik, Nova Scotia		20[3]	Acidified rivers. Decreased pH in lakes over a 20-30 year period.
Observed at BWCA	15[4,5]	(14)[5]	Stream pH depression of 0.8 units measured.
Observed at Kenora	12[5,6]	(11)[6]	No apparent detrimental effects.
Indicated Minimum Effects at Two Tolerances			
Almer/Dickson model	≅16		To protect moderately sensitive waters.
Almer/Dickson model	≅10		To protect the most sensitive waters.
Henriksen Nomograph		≅20*	To protect lakes above 50 μeq/l cations, i.e., the majority of sensitive lakes.
Henriksen Nomograph		≅12*	To protect the most sensitive waters.
C.D.R. estimate for Nova Scotia Rivers	≅7		To restore rivers to pH 5.8.
C.D.R. estimate for Nova Scotia Rivers	≅20		To restore rivers to pH 5.3.
pH Depression Relationship		≅10	Required to prevent detrimental pH depression in streams and lakes.

*Assuming 1 meter of rain per year, since the model prediction is based on precipitation pH.

[1]Likens et al. (28, 30); [2]Johannes et al. (11); [3]Harvey et al. (20); [4]Raglund and Wilkening (31); [5]Glass and Brydges (9); [6]Glass and Loucks (14).

sensitivity is a function of geochemical processes that vary greatly among watersheds; abatement to a level that protects the single most sensitive stream or lake in the U.S. or Canada is probably not feasible. When aquatic systems are inventoried according to defined sensitivity measures, a cutoff point may then be determined for the agreed on loading tolerances.

DISCUSSION AND CONCLUSIONS

Despite the qualitative structure of the model available for expressing linkages in the mediation of acidic inputs, use of the model has served a critical role in assuring coverage of relationships and systematic evaluation of the significance of otherwise fragmented evidence. However, more quantitative evaluation of this evidence in the context of mechanistic models is needed if the empirical models are to be understood.

Field validation of mechanistic cause/effect linkages in these systems will be extremely difficult unless a rigorous methodology like sequential hypothesis testing is used. Such a methodology requires a well-designed and comprehensive data base that was not available for the Phase II Studies (1). Ultimately, hypotheses that are not rejected can be summarized and incorporated as predictive or summarative components of the mechanistic models, representing an increasing integrative statement of relationships amenable to further testing. Such models function as semi-predictive tools and research summaries which are suitable for at least comparative evaluation of prospective responses under various pollutant control scenarios.

For the present, sequential examination of the information base in the context of a qualitative model leads to the conclusion that most of the response processes are reasonably well understood. This is true despite the

Reproduced from best available copy.

absence of quantitative formulations for each process. The understanding of these response processes plays an important role in concluding that the empirical models are reasonable quantitative formulations for the regions and conditions they represent, and that the relatively consistent results are more than fortuitous. These projections of ecosystem tolerances to increased chemical deposition appear likely to be the focus of much additional research and discussion. Judgment of their reliability at this time is, in many ways, a judgment of the synthesis methodology being used.

Reproduced from best available copy.

REFERENCES

1. U.S.-Canada Memorandum of Intent on Transboundary Air Pollution. 1981. Impact Assessment Working Group I, Phase II Interim Working Paper. Washington, D.C. and Ottawa, Ontario, Canada. 345 pp. in 7 sections.

2. Loucks, O. L. 1975. Models linking land-water interactions around Lake Wingra, Wisconsin. pp. 53-63 In: A. D. Hasler (ed.) Coupling of Land and Water Systems. Ecological Studies 10. Springer-Verlag, New York.

3. Loucks, O. L., R. W. Usher, D. Rapport, W. Swanson, and R. W. Miller. 1981. Assessment of Sensitivity Measures for Evaluating Resources at Risk from Atmospheric Pollutant Deposition. Report APP-0080-1980a to the U.S. Environmental Protection Agency, ERL-Duluth, by The Institute of Ecology, Indianapolis. 87 pp.

4. Holcomb Research Institute (HRI). 1976. Environmental Modeling and Decision Making: The United States Experience. A report to the Scientific Committee on Problems of the Environment. Praeger Publishers, New York. 152 pp.

5. Lettenmaier, D. P., and J. E. Richey. 1979. Use of first-order analysis in estimating mass-balance errors and planning sampling activities. pp. 80-106 In: E. Halfon (ed.) Theoretical Systems Ecology: Advances and Studies. Academic Press, New York. 516 pp.

6. Innis, G. S. 1975. The use of a systems approach in biological research. pp. 369-392 In: G. E. Dalton (ed.) Study of Agricultural Systems. Applied Science Publishers, Ltd., London. 441 pp.

7. Battelle. 1980. Overview Section: Summary, Objections, Conclusions and Project Organization. A research report to the American Electric Power Service Corporation. Battelle Laboratories, Columbus, Ohio. 21 pp.

Reproduced from best available copy.

8. Council on Environmental Quality (CEQ). 1981. National Acid Precipitation Assessment Plan. U.S. Interagency Task Force on Acid Precipitation. Washington, D.C. 130 pp.

9. Glass, G. E., and T. G. Brydges. 1981. Problem complexity in predicting impacts from altered precipitation chemistry. Intl. Conf. on Acid Precipit. Impacts, Cornell University, Ithaca, New York, August 1981.

10. Loucks, O. L., R. W. Miller, and T. V. Armentano. 1982. Regional Assessment of Aquatic Resources at Risk from Acidic Deposition. Final Report to the Office of Technology Assessment, U.S. Congress. The Institute of Ecology, Indianapolis (In press).

11. Johannes, A. H., E. R. Altwicker, and N. L. Clesceri. 1981. Characterization of Acidic Precipitation in the Adirondack Region. EPRI EA-1826, Project 1155-1, Final Report by Rensselaer Polytechnic Institute, Troy, New York, for Electric Power Research Institute, Palo Alto, California.

12. Pfeiffer, M. H., and P. J. Festa. 1980. Acidity status of lakes in the Adirondack region of New York in relation to fish resources. Progress Report. New York State Dept. of Environmental Conservation. Albany, New York.

13. Eilers, J. M., et al. 1979. Acid precipitation investigation for northern Wisconsin. Progress Report. Wisconsin DNR, Rhinelander; and U.S. EPA-ERL Duluth, Minnesota.

14. Glass, G. E., and O. L. Loucks (eds.). 1980. Impacts of airborne pollutants on wilderness areas along the Minnesota-Ontario border. EPA-600/3-80-044. U.S. Environmental Protection Agency, ERL-Duluth. 186 pp.

15. Arnold, D. E. 1981. Vulnerability of lakes and streams in the middle Atlantic states to acidification. Interim Progress Report. Cooperative Fisheries Research Unit, Pennsylvania State University, University Park, Pennsylvania. [As cited in (1).]

16. Haines, T. A. 1981a. Effects of acid rain on Atlantic salmon rivers and restoration efforts in the United States. In: Proceedings of the Conference on Acid Rain and the Atlantic Salmon. Portland, Maine, 1980. International Atlantic Salmon Foundation Special Publication Series No. 10. IASF, New York.

17. Hendry, G. R., J. N. Galloway, S. A. Norton, C. L. Schofield, P. W. Shaffer, and D. A. Burns. 1980. Geological and hydrochemical sensitivity of the eastern United States to acid precipitation. EPA-600/3-80-024. U.S. EPA-ERL, Corvallis, Oregon. 99 pp.

18. Klick, T. A., and C. W. Threinen. 1968. Surface Water Resources of Jackson County. Wisconsin Dept. of Natural Resources, Madison, Wisconsin.

19. Andrews, L. M., and C. W. Threinen. 1966. Surface Water Resources of Oneida County. Wisconsin Dept. of Natural Resources, Madison, Wisconsin.

20. Harvey, H. H., R. C. Pierce, P. J. Dillon, J. P. Kramer, and D. M. Whelpdale. 1981. Acidification in the Canadian Aquatic Environment: Scientific Criterion for an Assessment of the Effects of Acidic Deposition on Aquatic Ecosystems. Nat. Res. Coun. Canada Report No. 18475. 369 pp.

21. Loucks, O. L. 1980. Acid Rain: Living Resource Implications and Management Needs. Proceedings, Forty-Fifth North American Wildlife Conference. Wildlife Management Institute, Washington, D.C. pp. 24-37.

22. Almer, B., W. Dickson, C. Ekstrom, and E. Hornstrom. 1978. Sulfur pollution and the aquatic ecosystem. pp. 271-311 In: J. O. Nriagu (ed.) Sulfur in the Environment. Part II. Ecological Impacts. J. Wiley and Sons, Inc., New York. 568 pp.

23. Henriksen, A. 1980. Acidification of freshwaters - a large scale titration. pp. 68-74 In: D. Drabløs and A. Tollan (eds.) Ecological Impact of Acid Precipitation. Proceedings of an international conference. SNSF Project, Norway. 383 pp.

24. Thompson, M. E. 1982. The cation denudation rate as a quantitative index of sensitivity of eastern Canadian rivers to acidic atmospheric precipitation. Water Air Soil Pollut. 18: 215-226.

25. Crisman, T. L., and P. L. Brezonik. 1980. Paper for presentation to 73rd Annual Meeting Air Pollut. Contr. Assoc., Montreal, Quebec, Canada.

26. Lewis, W. M., Jr., and M. C. Grant. 1979. Changes in the output of ions from a watershed as a result of the acidification of precipitation. Ecology 6: 1093-1097.

27. Likens, G. E., F. H. Bormann, R. S. Pierce, J. S. Eaton, and N. M. Johnson. 1977. Biogeochemistry of a Forested Ecosystem. Springer-Verlag, New York, Inc. 146 pp.

28. Wright, R. F., and E. Snekvik. 1978. Acid Precipitation: Chemistry and fish populations in 700 lakes in southernmost Norway. Verh. Int. Verein. Limnol. 20: 765-775.

29. Likens, G. E., F. H. Bormann, and J. S. Eaton. 1980. Variations in precipitation and streamwater chemistry at the Hubbard Brook Experimental Forest during 1964 to 1977. pp. 443-464 In: T. C. Hutchinson and M. Havas (eds.) Effects of Acid Precipitation on Terrestrial Exosystems. Plenum Press, New York. 654 pp.

Reproduced from best available copy.

30. Raglund, K. W., and K. E. Wilkening. 1981. Relationship between mesoscale acid precipitation and meteorological factors. In: F. M. D'Itri (ed.) The Effects of Acid Precipitation on Ecological Systems in the Great Lakes Region. Proceedings of a conference. Ann Arbor Science Publishers, Ann Arbor, Michigan (In press).